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Original research article

Impacts of land-use change on sacred forests at the landscape scale



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ABSTRACT

Sacred forests often exist as isolated patches of natural forest even after conversion of the surrounding matrix to different forms of land-use. This study set out to: (1) evaluate land-cover changes and patch fragmentation in a landscape containing sacred and non-sacred forest patches over 15 years and (2) compare the effects at an individual patch level between sacred and non-sacred forests. Past changes in area and patch fragmentation of land cover classes and individual forest patches in the Gamo Highlands, Ethiopia, were assessed using maximum-likelihood classification of LANDSAT images. Large changes in land-cover occurred during 1995–2010, with 109.4% increase in area of farm and settlement and 36.6% decrease of forest area, with a decrease in number of forest patches by 16.1%, mean size by 26.8%, edge density by 29.1% and shape index by 13.3%. While all four individually studied non-sacred forests decreased in size over this period only four of the six individual sacred forests patches showed reduction in area. Forest patches with sacred status had greater protection by local communities than non-sacred forests in the Gamo Highlands. However, their small size and increasing edge density indicate high vulnerability, especially if an erosion of traditional cultural values reduces their protection.

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1. Introduction

Deforestation and degradation of wooded habitats due to anthropogenic activities (especially land-use change) are among the major contributors to current global climate change and biodiversity reduction (Foley et al., 2005; Echeverría et al., 2007; Zanella et al., 2012). Human population growth pressures are expanding the area of land-uses such as agriculture and settlement into natural habitats in all parts of the world to meet the demand for food and housing (Lambin et al., 2003; Kabba and Li, 2011). These land-use changes have led to deforestation, further aggravating fragmentation of remaining forest habitats (FAO, 2003; Ellis-Cockcroft and Cotter, 2014; Riutta et al., 2014). In response to this, large investments have been made in the establishment of nature reserves across the world to preserve large pristine areas. The establishment of these reserves has sometimes not been successful as the initiatives are often politically driven and aspire to achieve environmental benefits without the involvement of immediate users or local communities (Brown, 2003; Bhagwat and Rutte, 2006; Khan et al., 2008; Pullin et al., 2013). On the other hand, many local communities conserve forest patches in their habitation area in the form of sacred forests for cultural purposes (Claudia, 2008; Uyeda et al., 2014). In addition to their cultural significance, these forests are important for the conservation of species useful to local people (Wadley and Colfer, 2004) and of biodiversity conservation importance (Mgumia and Oba, 2003; Anderson et al., 2005; Bhagwat and Rutte, 2006;

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Rim-Rukeh et al., 2013; Tamalene et al., 2014). Consequently, despite increased pressures, they have coexisted with human populations for centuries where they are under the custody of whole communities or sufficiently influential religious leaders. They may therefore serve both local resource needs and international conservation goals (Swamy et al., 2003; Wadley and Colfer, 2004; Kokou et al., 2006; Ormsby and Bhagwat, 2010; Umazi et al., 2013).

Protection of sacred forests that remain in landscapes dominated by agriculture is of critical importance in many countries due to the past rapid loss of natural habitat as a result of land-use change. The value of these forest patches for biodiversity conservation has gained increasing attention through recent studies (Bhagwat et al., 2005; Khumbongmayum et al., 2006; Ambinakudige and Sathish, 2009; Lehouck et al., 2009; Echeverría et al., 2012; Sponsel, 2012; Gao et al., 2013; Tamalene et al., 2014). The continuing protection of these remnant sacred forests by traditional custodians, due to the high local value of the associated culture, has recently been recognized at the international level (Ramanujam and Kadamban, 2001; Ormsby, 2011; Corrigan and Hay-Edie, 2013; Ormsby, 2013). Sacred forests are mentioned in the Convention on Biodiversity (CBD, 1992), the Sacred Site Programme proposal of the United Nations Educational, Scientific and Cultural Organization (UNESCO, 1996) and in the Sacred Natural Site (SNS) management guidelines published by the International Union for Conservation of Nature (Wild and McLeod, 2008). These, however, tend to focus only on the biodiversity and ecological importance of the sacred forests. They do not provide a detailed analysis of the threat posed by land-use activities in the surrounding matrix. Furthermore, there is a lack of spatio-temporal characterization of sacred forests at the patch level or quantification of the rate of change in surrounding land-use/land-cover at the landscape scale, which are crucial indicators of the current status of sacred forests, trends in the recent past and requirements for their future conservation.

Ethiopia has many sacred forests under the stewardship of indigenous communities. They are particularly common in south-west Ethiopia, where they have been protected by strong local beliefs in their status as sacred sites and consequent strict application of religious rules for their protection (Desalegn, 2007; Berhane-Selassie, 2008). The Gamo Highlands are recognized as a particularly important area for these sacred forests: our vegetation survey within sacred forest patches showed high species diversity and abundance in comparison with non-sacred forests in the same region (Desalegn, 2012). They also contain species endemic to Ethiopia, globally threatened species (IUCN, 2010) and species endemic to afro-montane forests (Desalegn, 2012). Their size is highly variable (from 0.6 ha to 500 ha) and they are generally surrounded by a matrix of agricultural fields and other land uses associated with human habitation. Although research employing remote sensing and GIS has quantified the land-use and land-cover changes of the area (Desalegn, 2007; Teshome, 2012a,b), no study has been focused at the scale of sacred forest patches. The objectives of this study are therefore to: (1) evaluate the impact of land-use change over the past 15 years on the spatial properties of areas of different land-cover classes across the landscape of the Gamo Highlands and (2) characterize the changes associated with fragmentation occurring to individual patches of sacred and non-sacred forests within this landscape over the same period.

2. Study area

The Gamo Highlands are a section of the south-west Ethiopian highlands, and are located in the GamoGofa Administrative Zone in the Southern Nations, Nationalities and Peoples Regional State (SNNPRS) (Fig. 1). GamoGofa Zone Administration consists of 15 woredas (small administrative unit equivalent to district) of which Gamo Highlands comprise larger portion. According to the population census of 1994, the total population inhabiting the Gamo highlands was estimated to be 700,000 (Freeman, 2002) and this had increased to more than 1 million in 2007 (CSA, 2007). People's livelihoods in the highlands mainly depend on subsistence farming of generally less than 1 hectare (Samberg et al., 2010).

The Gamo Highlands are located on the western escarpment of the Great Rift Valley between 5°53.8'17.52" to 6°26'22.97" N and 37°10'35.13" to 37°42'31.89" E. The topography of the highlands is characterized by steep slopes, up to undulating plateaus with gentle slopes, as well as detached steep-sided hills and valleys (Desalegn, 2007; Samberg et al., 2013a,b). The elevation of the highlands rises abruptly from 1183 m in the Maze lowlands to the west and from 1200 m at Lake Abaya and Chamo in the east to the central ridges with a maximum elevation of 3500 m at the summit of Mount Guge. The climate of the area is characterized by a bimodal rainfall pattern with high intensity rainfall from June to September and low intensity from February to April. The mean annual rainfall ranges from 500 mm in the lowlands to 1200 mm in the highlands, and the temperature varies from 25 °C in the lowlands to 10 °C in the highlands (Federal Democratic Republic of Ethiopia, 2000).

3. Methods

3.1. Land-cover data and analyses

Cloud-free LANDSAT Thematic Mapper (TM) images of an area of 183 × 183 km covering the majority of the Gamo Highlands were obtained for the years 1995 and 2010. They gave a total sample area in the Gamo Highlands of 66,765 ha. A maximum-likelihood classification of the data of each TM image was carried out to map land cover for mapping units of 0.5 ha. The classification was aggregated into four cover classes of: cultivated land and settlement; forest; open pasture land; wooded grassland. Classification accuracy was verified to 78% for 1995 and 85% for 2010 in ERDAS 9.1 (ERDAS, 2008) using 270 ground control points (GCP) obtained during field survey. The contour map and administrative boundary vector

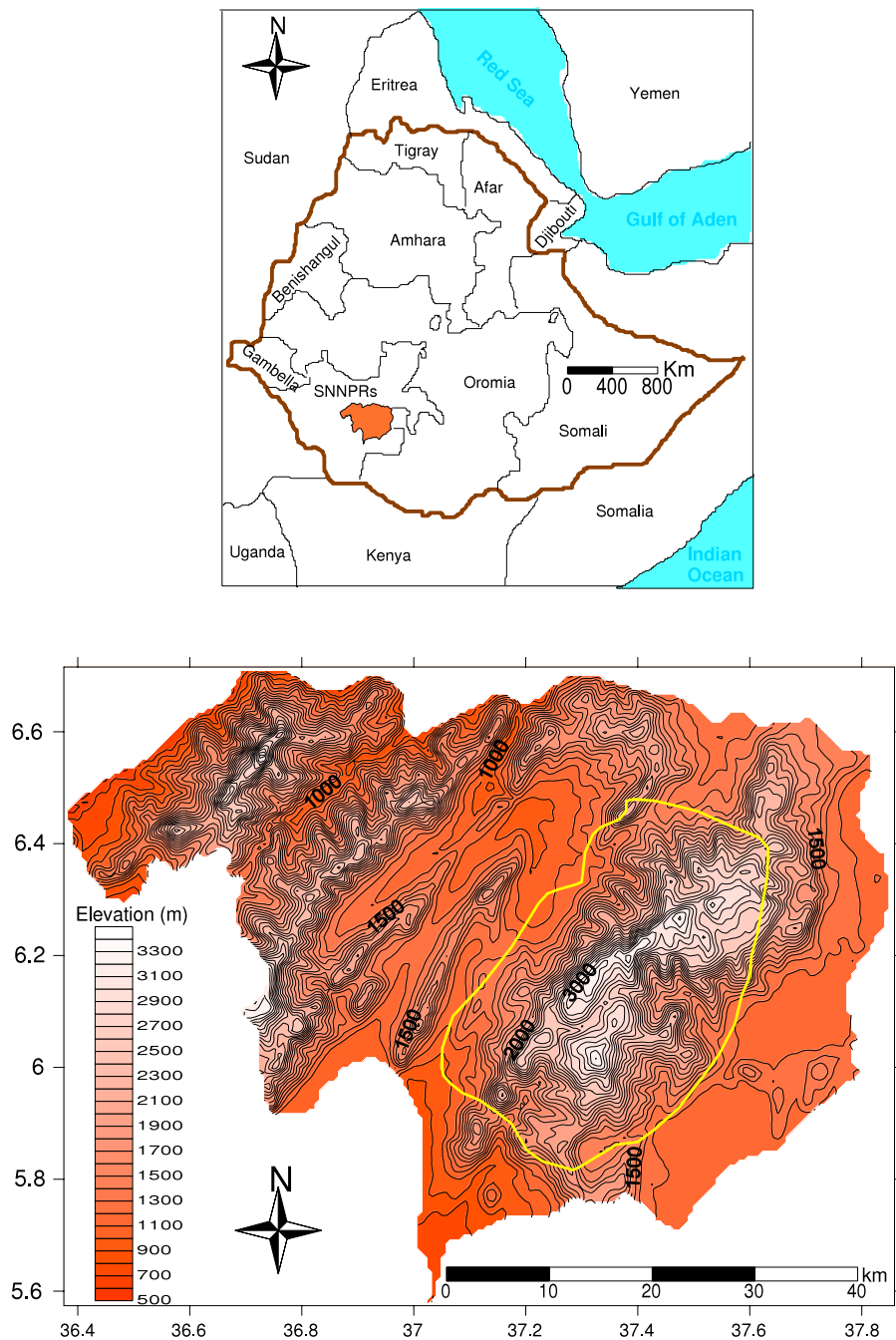


Fig. 1. Study location. (A) Ethiopia (with thick line international boundary), the states of Ethiopia, showing the Southern Nations, Nationalities and Peoples Regional State (SNNPRs) obtained from the Ethiopian Map Agency and, within it shown in brown, the GamoGofa Administrative Zone in south-west Ethiopia in which the study was carried out, (B) shaded topographic map of the GamoGofa Administrative Zone including the Gamo Highlands study area shown as a yellow polygon. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

maps of the area produced by the Ethiopian Mapping Agency were also used to clip the area of interest to determine its position and ensure that the specific area covered by the two images was the same.

The extent of land-cover types in each map and their reduction or gain in total area (1995–2010) in the whole 66,765 ha were evaluated by computing the proportion of each land-cover class. The Class area (CA) for each of these four classes were compared to define landscape composition (the percentage of the area comprised by each land-cover class [Weng, 2007](#)). This is important for comparing among landscapes of varying sizes ([Gergel and Turner, 2002](#)). Individual patches of each land-cover type were defined by eight-neighbour rules (cells with the same edge that share a common corner are

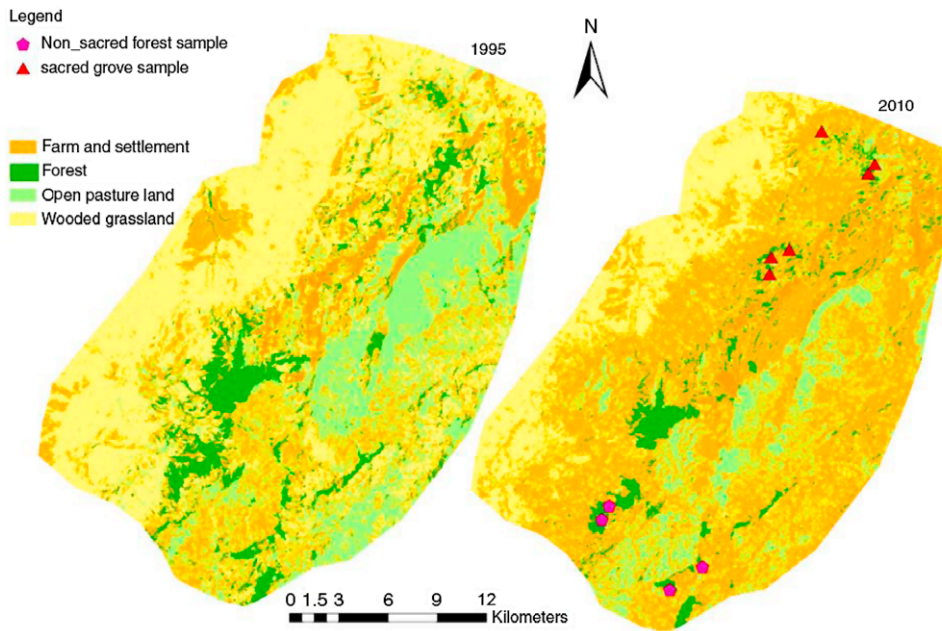


Fig. 2. Land-cover classification of the Gamo Highlands study area in Ethiopia in 1995 (left) and 2010 (right) showing land cover change and the location of the six sacred forests (triangles) and four non-sacred forests (pentagons).

considered part of the same patch) and adjacent pixels belonging to the same land-cover class of the classified images and delimited by patch boundary. Where two adjacent patch polygons had the same land cover the boundary between the two was dissolved in order to reduce the number of patches to a realistic minimum (Rempel, 2012). Fragmentation was then evaluated for each map by quantifying the number of patches (PN), mean patch size (MPS), mean shape index (MSI), edge density (ED), Shannon diversity index (SDI) and Simpson evenness index (SEI). The number of patches (Nump) was used to quantify the number of individual patches of each of the four land-cover classes in the landscape and its increase or reduction indicates fragmentation or loss of habitat. Mean patch size (MPS) measures the average area of all patches for each class (Gergel and Turner, 2002). It is the major index of natural habitat pattern that affects biodiversity conservation and species composition and diversity (McGarigal and Marks, 1995) therefore it has particular relevance as an indication of patch viability for the forests. The mean shape index (MSI) is based on the perimeter-to-area ratio of each patch, and indicates the influence of edge effects within the patch, therefore this is of particular interest for forest patches surrounded by a matrix of farm or pasture land. The value varies between 1 and 2; low values are derived when a patch has compact rectangular form with a relatively small perimeter-to-area ratio. Edge Density (ED) is a measure of the total length of the edge of all of the patches in each land-cover class divided by the total landscape area. A large value of edge density indicates a high level of human disturbance and fragmentation of the class. The Shannon diversity index is defined as $SHDI = 1 - \sum_{i=1}^N p_i \times \ln p_i$ where N is the number of land cover types and P_i the proportional abundance of the i th type. The value ranges from 0 to infinity; it increases where the number of land cover types increases (Nagendra, 2002). The Simpson evenness index is defined as $SIDI = 1 - \sum_{i=1}^N P_i \times P_i$ with the value range from 0 to 1; it increases where the equitability of distribution of land amongst the various cover types increases. The selection of these landscape metrics was based on their high frequency of use in landscape analysis by different authors (Bierwagen, 2007) as well as their close fit with the objectives of this study. The metrics were quantified using Patch Analyst 4.2 (Elkie et al., 2012) at landscape, class and patch levels for each of the two years. Patch Analyst was chosen because it provides an integrated GIS environment for spatial analysis.

Forests in the Gamo highlands are under a range of different governance regimes. Some forests are considered sacred and protected accordingly by community and religious leaders while others have no sacred status and are generally subjected to a much wider range and intensity of uses by the surrounding communities. To compare land-use change impacts between these two governance types, six sacred forests (Ula, Gufae, Qimme, Tele, Osha-Ocha and Akasie) were selected that were accessible and whose sacred status could be confirmed through interviews of local community members. They occurred within an elevation range of 2022–2438 m a.s.l. and surrounded by two predominant landscape matrix (agricultural field and grazing land). Then the available non-sacred forests from the few remaining in the Gamo Highlands that best matched the elevation, physical site characteristics, local socio-economic conditions and similar landscape matrix of the six selected sacred forests were chosen; they were four in number (Sora, Shoa, Ochee and Dhule, Fig. 2). The changes in size, shape and edge density over the 1995–2010 period were then compared between these sets of patches of the two forest types. Due to their difference in patch area between the two forest types, the shape was calculated by dividing patch perimeter by the

Table 1

Land-cover change of the Gamo Highlands study area (1995–2010).

Year/period	1995		2010		1995–2010 change	
Land-cover classes	Area (ha)	%	Area (ha)	%	Area (ha)	%
Farm and settlement	19,255	28.84	40,329	60.4	21,074.0	+109.4
Forest	7,216	10.81	4,573	6.8	–2643.0	–36.6
Grazing land	16,878	25.28	10,390	15.6	–6488.0	–38.4
Wooded grassland	23,415	35.07	11,472	17.2	–11,943.0	–51.0
Total	66,764	100.00	66,764	100.0	0	0

square root of patch area, adjusted using a square standard to remove area effect; and edge density standardize edge to a per unit area to correct for perimeter bias (McGarigal and Marks, 1995).

4. Results

4.1. Landscape composition

There was a large change in land cover in the study area due to human land-use change in the 15-year period, 1995–2010 (Fig. 2, Table 1). Wooded grassland was the dominant and most connected land-cover class in 1995, occupying 35.1% of the study area, however almost half of its area had been converted to farm and settlement land cover by 2010 (Fig. 3(a)). Farm and settlement was second largest in area (28.8%) in 1995, then it more than doubled (109.4% increase) to become the dominant land cover by 2010. Open pasture land was 25.3% of the area sampled in 1995 and was largely restricted to high elevation areas which include the alpine grassland habitats. This land-cover type reduced to 15.6% of the total area by 2010, a loss of 6488 ha. Forest covered just 7216 ha (10.8%) of the area in 1995 and was reduced to 4573 ha (6.8%) in 2010 (Fig. 3(a)). In summary, between 1995 and 2010 wooded grassland lost 51.0% of its area, open pasture 38.4% and forest 36.6%, all predominantly to farm and settlement land (Table 1).

4.2. Fragmentation at patch level for each land cover class

There was a decrease in the number of patches of all four land-cover classes between 1995 and 2010 indicating a simplification of the pattern of land cover (Fig. 3(b), SI Table 1). Forests had the smallest reduction in patch number (from 1718 to 1441 (16.1%, or 18.5 patches per year). In 1995 individual patches of farm and settlement land were often still interspersed by land of the three other land-cover classes having a mean patch size of only 1.7 ha (Fig. 3(c), SI Table 1). By 2010 the patches of farm and settlement land had often coalesced through conversion of the intervening land-cover, increasing their mean patch size more than five-fold to 9.9 ha. During this process there were decreases in both patch number and mean patch area of all three of the other land-cover classes. This simplification of the landscape structure was also reflected in the reduction in values of patch Mean Shape Index and Edge Density metrics for all four land-cover classes between 1995 and 2010 (Fig. 3(f) and (d), SI Table 1). The percentage reduction in edge density was greatest for wooded grassland and least for farm and settlement, whereas the reduction in mean shape index (indicating an increase in regularity of shape) was greatest for forest. The Patch Size Coefficient of Variation (PSCOV) value, indicating variation of patch size in the landscape, increased for farm and settlement land, whilst decreasing for the other three classes, especially open pasture land and wooded grassland (Fig. 3(e)).

4.3. Patch fragmentation at the landscape level

When fragmentation is assessed at the landscape level (which sums across all four land-cover classes) the same trends of simplification and reduction in heterogeneity between 1995 and 2010 are apparent. The large increase (by 81.4%) in mean patch size is attributable to the loss of most small patches to other land-uses as indicated by the decrease in patch number by 44.3% and reduction in patch size coefficient of variation by 9.1% (Table 2). These trends resulted in a reduction in both the Shannon diversity and Simpson evenness indices for the landscape of 2010 compared with 1995 (Table 2). These landscape-level changes were also associated with changes in individual patch attributes with a large reduction in mean edge density. Edge density was reduced by 22.8% and mean shape index slightly.

4.4. Fragmentation of individual sacred forests and non-sacred forests at the patch level

Fragmentation was also assessed at the individual patch scale for the six selected sacred forests and four non-sacred forests. Change in patch size was very variable amongst the sacred forests, increasing in two but decreasing in the other four (Fig. 4(a)), resulting in a decrease in mean patch size of only 5.8% (SI Table 2). In contrast, all four non-sacred forests showed a large decrease in size with a decrease in the mean of 43.2% (Fig. 4(b)). Change in shape index between 1995 and 2010 was correlated with change in patch size and so was similarly variable amongst the six sacred forests with little overall change

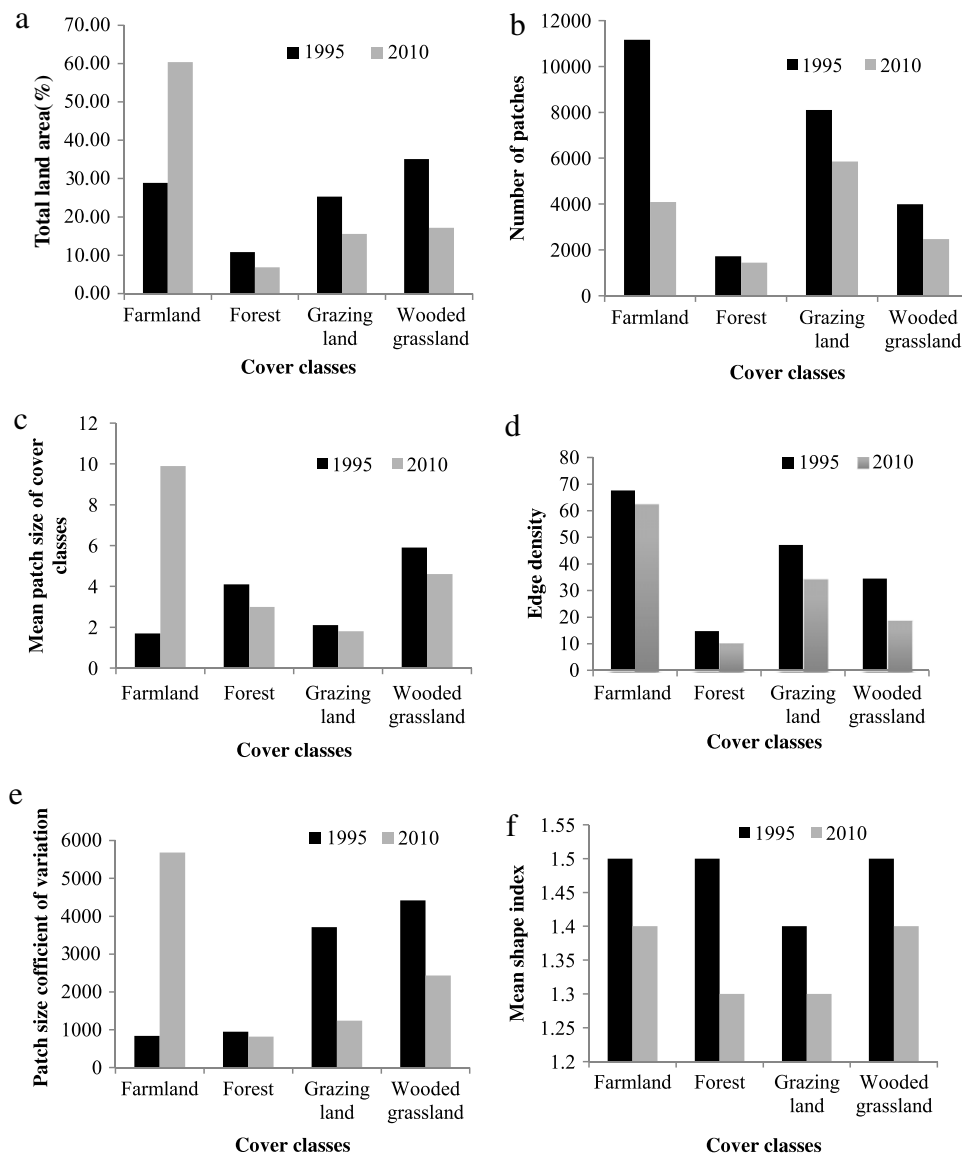


Fig. 3. Fragmentation of land-cover classes in the Gamo Highlands study area in 1995 and 2010: (a) land-cover of each land-cover class, (b) number of patches, (c) mean patch size (ha), (d) edge density (m), (e) patch size coefficient of variation, (f) mean patch shape index. Farmland comprises cultivated and settled land.

Table 2

Fragmentation metrics of all four defined land-cover classes combined in the Gamo Highlands study area in 1995 and 2010. Mean patch size (ha) (MPS); number of patches (NumP); patch size coefficient of variation (PSCOV); Shannon diversity index (SDI); Simpson's evenness index (SEI); mean shape index (MSI); edge density (ED); total area (ha) (TA).

Year	MPS	NumP	PSCOV	SDI	SEI	ED	MSI	TA
1995	4.3	25,003	6620.3	1.5	0.9	166.6	1.5	66,765
2010	7.8	13,926	6016.8	1.3	0.8	128.6	1.4	66,765

in the mean (a reduction of just 2.8%, Fig. 4(c), SI Table 2) and decreased in all four non-sacred forests (a reduction in the mean of 34.0%) indicating increased regularity of shape (Fig. 4(d)).

The mean edge density increased in all six sacred forests with a significant increase in the mean of 27.3% (Fig. 4(e) and (f), SI Table 2). In contrast, for the non-sacred forests edge density increased in two and decreased in two, resulting in a very similar mean for 1995 and 2010. In 1995 edge density was higher for all six sacred forests than for all four non-sacred forests; this was still the case for five of the sacred forests in 2010 (SI Table 2).

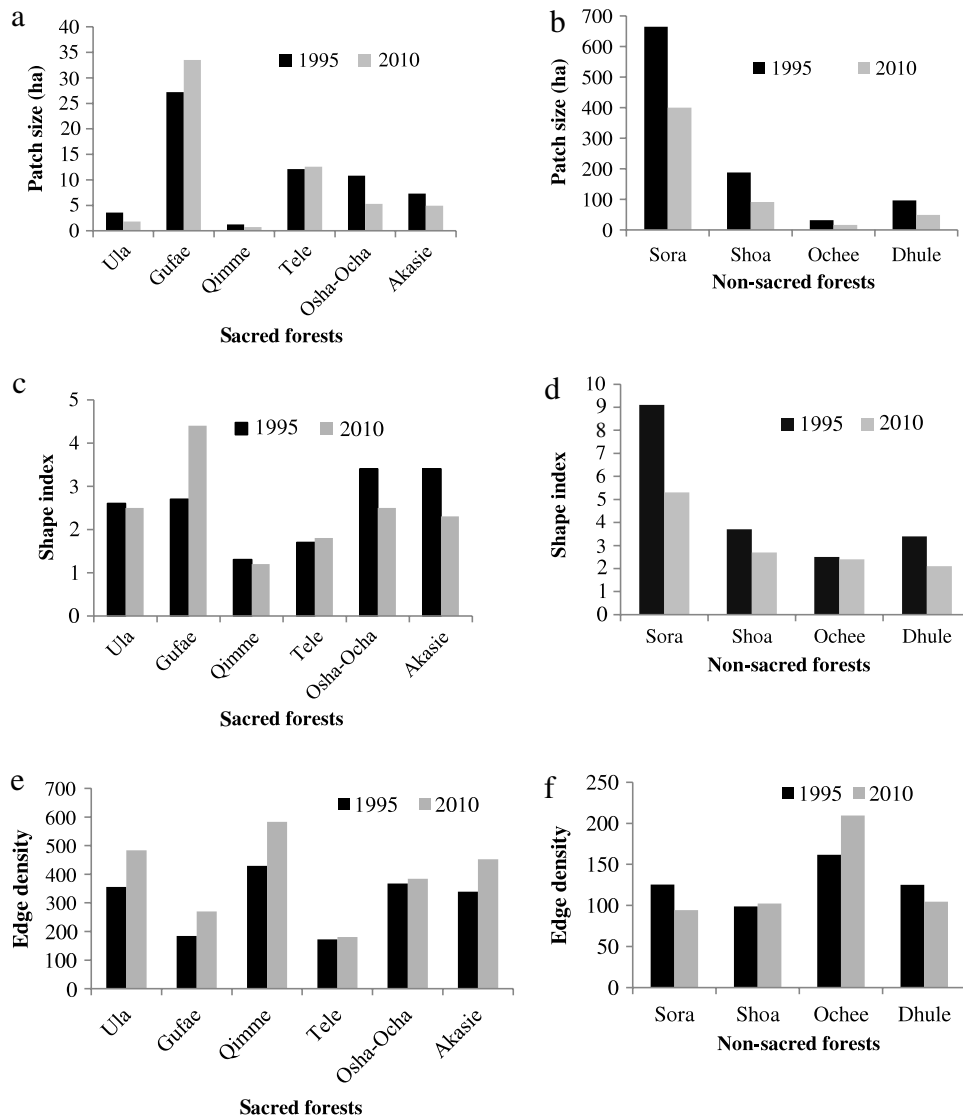


Fig. 4. Patch size (ha) (a and b), shape index (c and d) and edge density (e and f) of six sacred forests and four non-sacred forest patches in the Gamo Highlands of Ethiopia in 1995 and 2010.

5. Discussion

5.1. Land-use and land-cover change in the Gamo Highlands

Expansion of agricultural and settlement land-use greatly reduced the land-cover of natural habitats in the Gamo Highlands' landscape during the 1995–2010 study period, reflecting the trends reported across the tropics by [Lambin et al. \(2003\)](#). This has led to fragmentation (a reduction in the spatial relationship between natural habitats) as the landscape spatial pattern has changed ([Gustafson, 1998](#), [Turner et al., 2001](#), [Aguirre and Dirzo, 2008](#), [Lehouck et al., 2009](#), [Kabba and Li, 2011](#)). Up to 1995 the landscape was dominated by natural/semi-natural habitats of woodland, wooded grassland and open pastureland, forming the matrix for the comparatively small patches of household farms and settlements. Farms were usually located on gentle slopes and valley bottoms, while the steep slopes and flat plateaus were used as open communal pasture land. In contrast, by 2010 the area of farmland and settlement had expanded massively into land previously occupied by all three natural/semi-natural habitat types and now forms the matrix of most of the landscape except for one strip dominated by wooded grassland in the west and one dominated by open pasture to the east of the highlands ([Fig. 2](#)).

Evidence of an increase in fragmentation of habitat patches within this landscape is provided by the reduction in mean patch size ([Echeverría et al., 2007](#)). However, the large reduction in patch number of all three natural/semi-natural

habitat types (especially wooded grassland) shows that many have been diminished to destruction rather than remaining as fragments, so the best overall descriptor is attrition of the semi-natural land-cover types rather than fragmentation (Forman, 1995; Lindenmayer and Fischer, 2006). The large decrease in edge density indicates a major reduction in the spatial heterogeneity of the landscape (McGarigal et al., 2002). The reduction in patch number and Shannon Diversity Index both indicate reduction in the landscape's habitat richness (McGarigal and Marks, 1995). Together these changes are likely to have reduced landscape-level biodiversity (Lindenmayer and Fischer, 2006), indicating the urgent need for assessment of conservation priorities. Virtually all the remaining forest patches are now within a matrix of farm and settlement land. This combination of a great reduction in the area of forest patches (through encroachment for agriculture and settlement) and the severe fragmentation of the patches due to being surrounded by this matrix of intensively used land-cover is likely to pose a severe threat to the conservation of forest biodiversity in this landscape.

5.2. Changes in size, shape and edge density of sacred and non-sacred forests

Whereas the four studied non-sacred forest patches all showed a consistent pattern of attrition (area loss), the trend was more variable amongst the six sacred forests. The four smaller sacred forests all suffered a reduction in area and shape index, but the larger two increased. However, there was a consistent result of increase in edge density for all six sacred forests, indicating that they may all have suffered from disturbance and small-scale incursions around their edges; during ground survey all six were observed to be completely isolated within an agricultural matrix (as is indicated in Fig. 2).

These results indicate that forests in the Gamo Highlands which are not under the cultural protection of sacred status have suffered a high level of habitat loss and fragmentation. The situation for the sacred forests is more mixed. Gamo religious leaders and community elders still adopt traditional sacred values, cultural taboos, belief systems and bylaws to protect sacred forests, as has also been observed in many other societies around the world (Bhagwat and Rutte, 2006). However, in practice this is not effectively ensuring that all of them are conserved. As evidenced by the opinions of the traditional custodians of the six studied sacred forests during our interviews of them, and our observations during vegetation survey within the forests (Desalegn, 2012), many of the sacred forests are under threat, reflecting the complex economic, social and cultural changes currently taking place in the Gamo Highlands. The main physical threats to the sacred forests that we observed during field visits were grazing and some tree cutting. No active management by enrichment tree planting or restoration was observed.

6. Conclusions

The Gamo Highlands landscape experienced extensive conversion from natural/semi-natural to human-dominated land-use during the 1995–2010 period. This has greatly fragmented the remaining patches of natural/semi-natural habitats including forests, which are increasingly isolated within an agricultural landscape. The reduction in forest patch size and increase in edge density is likely to reduce greatly the value of many of the remaining patches for biodiversity conservation and may accelerate their complete destruction. This has led to an increase in the relative importance of the sacred forests for conservation as they have suffered a lower rate of area loss than the non-sacred forests. However, their capacity to provide this benefit is also highly threatened by fragmentation and the effect of the surrounding farmed and settled matrix. There is also a risk of positive feedback in physical loss of sacred forests and decline in local cultural values that have previously protected them. This provides a clear example of the synergy between the provision by forests of biodiversity conservation and cultural ecosystem services. Support for the cultural values associated with sacred forests has a crucial role to play in the long-term sustainable conservation of these habitats.

In order to reverse this spiral of decline it will be important for the governance of sacred forests by their traditional custodians to be supported and not undermined by the national legal framework and by governmental and non-governmental organizations. Working together to better protect the sacred forests, these institutions should enact management measures including (i) creating buffer zones around sacred forests with a high cover of the tree species occurring in the sacred forests in order to reduce the effects of high edge density, (ii) increasing the sustainability and habitat heterogeneity of agricultural land use in the surrounding matrix, e.g. through agroforestry practices, and (iii) habitat restoration within the sacred forests in order to enrich the populations of threatened species. Where browsing by cattle is a particular local threat to sacred forests, the construction of perimeter walls or fences can play a valuable role in their protection. The involvement of multiple institutions will be essential to ensure that management is not carried out for each sacred forest in isolation, but is planned at a landscape scale (Lindenmayer and Franklin, 2002). The negative effect of the small size and isolation of many remaining sacred forests presents a threat to the viability of many species' populations. This can be countered by establishment and protection of a network of other forest patches within the landscape providing the habitat to support metapopulation of forest species (Hanski, 1998).

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Appendix A. Supplementary data

Supplementary material related to this article can be found online at <http://dx.doi.org/10.1016/j.gecco.2014.12.009>.

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